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Review

Review of LCA studies of solid waste management systems – Part II: Methodological guidance for a better practice

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ABSTRACT

Life cycle assessment (LCA) is increasingly used in waste management to identify strategies that prevent or minimise negative impacts on ecosystems, human health or natural resources. However, the quality of the provided support to decision- and policy-makers is strongly dependent on a proper conduct of the LCA. How has LCA been applied until now? Are there any inconsistencies in the past practice? To answer these questions, we draw on a critical review of 222 published LCA studies of solid waste management systems. We analyse the past practice against the ISO standard requirements and the ILCD Handbook guidelines for each major step within the goal definition, scope definition, inventory analysis, impact assessment, and interpretation phases of the methodology. Results show that malpractices exist in several aspects of the LCA with large differences across studies. Examples are a frequent neglect of the goal definition, a frequent lack of transparency and precision in the definition of the scope of the study, e.g. an unclear delimitation of the system boundaries, a truncated impact coverage, difficulties in capturing influential local specificities such as representative waste compositions into the inventory, and a frequent lack of essential sensitivity and uncertainty analyses. Many of these aspects are important for the reliability of the results. For each of them, we therefore provide detailed recommendations to practitioners of waste management LCAs.

Contents

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1. Introduction

Anthropogenic activities result in the generation of waste, which may cause significant damages to human health (public and occupational health), ecosystems and natural resources (Vergara and Tchobanoglous, 2012). As environmental sustainability is increasingly recognised and included in the public agenda, the sector of waste management is expected to minimise its impacts (EC, 2011; Hoornweg and Bhada-Tata, 2012). The traditional consideration of waste as a pollution has progressively shifted towards a new perspective, in which waste is regarded as a resource that could support societies to become more sustainable. For example, the energy recovered in certain thermal processes can avoid the generation of energy services via conventional technologies. Likewise, the reuse or recycling/downcycling of certain waste materials, e.g. metals, can save the production of virgin materials (UNEP, 2011).

In this setting, waste management policies have developed and frameworks have emerged to allow identifying the most adequate strategies. The prime example of such an initiative is the EU Waste Framework Directive (EU Directive 2008/98/EC) that establishes a 5-step waste hierarchy, legally posing in decreasing priority order the waste prevention, preparation for reuse, recycling, other recoveries, e.g. energy recovery, and disposal. Several of these strategies and policies have called for the use of life cycle assessment (LCA), which allows quantifying all relevant environmental impacts of a system stemming from its entire life cycle, thus aiming for a holistic perspective. But how has LCA been applied to solid waste management systems (SWMSs) until now? Taking the entire field as a whole, what key learnings and improvement potentials can be identified?

To address these questions, we performed a critical review of 222 studies, originating from 216 scientific articles and 15 public reports. The extensive scope of the review gives it a comprehensiveness that we believe is unmatched in previous works. In part I of this study (Laurent et al., 2013), in addition to detailing these scopes and approaches pertaining to the review itself, we analysed the studies in the broad perspective of global solid waste management. Results showed that LCA applications have largely been limited to developed countries, hence indicating that a number of environmental problems specific to waste management in developing countries, e.g. occupational health impacts from informal collection and recycling, have not been investigated. Waste prevention, which has the highest priority in the waste hierarchy, was also identified as suffering from a lack of consistent, operational methodology within an LCA context. An analysis of the findings from selected high quality studies demonstrated the problems that might arise when generalising LCA results and defining intrinsic rankings of waste management alternatives as that materialised by the waste hierarchy. In the assessment of SWMS, the local context, which determines highly-variant parameters, such as waste composition or energy supply mix, has a strong influence on what the optimal strategy is. Through its deeper understanding of SWMS, a well-performed LCA will capture those local specificities. Our results thus suggested that decision- and policy-makers should preferably rely on context-specific LCA studies, and use generalised results or the widely accepted waste hierarchy as such with caution. For full details and discussions of the above points, the reader is referred to Laurent et al. (2013).

In this sequel, we take one step further by addressing the reliability and consistency of the LCA results that are strongly dependent on a proper application of the LCA methodology. A number of studies have conducted reviews on LCA methodological practice in the field of SWMS (e.g. Cleary, 2009), but, because they all start by conducting a pre-selection of studies, none of them captures the overall practice in the field. The comprehensiveness of our review scope enables us to bridge this gap. Primarily targeting LCA practitioners in the field of waste management, we thus aim to take each major methodological step within the goal definition, scope definition, inventory analysis, impact assessment and interpretation, and (1) review how LCA has been applied in practice; (2) identify potential misuses and misunderstandings; and (3) provide specific recommendations to ensure a proper application of LCA in compliance with the ISO 14044 standards (ISO, 2006) and the more recent framework from the International Reference Life Cycle Data System (ILCD) Handbook (EC, 2010a, 2011).

2. Methods and materials

2.1. Review characteristics

The identification of the 222 studies, the criteria for inclusion of studies, the developed review scheme and the evaluation of the studies are fully documented in Laurent et al. (2013), and the reader is referred to this source for detailed information.

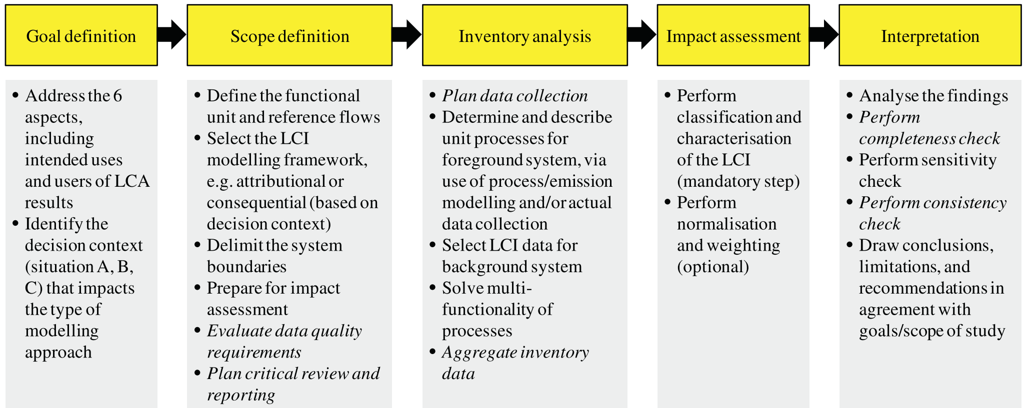

Fig. 1. Major LCA methodological steps (based on EC, 2010a, 2011). Steps marked in italic are not addressed in this paper.

In short, the retained studies are those that (1) have a focus on solid waste, which is regarded as neither water nor airborne (Christensen, 2011); (2) only encompass solid waste management systems, thus disregarding studies including upstream activities prior to waste generation (exception of waste prevention studies) and post-waste management system aspects; (3) include an impact assessment of more than one impact category, thus disregarding studies solely assessing climate change impacts; (4) are fully published in English language. Studies meeting all four conditions were included in the review. The adopted approach to retrieve the LCA studies in scientific literature consisted of screening a list of 15 relevant scientific journals and cross-checking the cited and citing literature for the articles published after 2009 (stopping the collection of studies at mid-year 2012). This approach is believed to bring representativeness in the mapping of the past LCA practice as related in the scientific literature. On the other hand, comprehensiveness could not be guaranteed for technical reports due to the searching methodology and the English language criteria.

2.2. Analysis of past LCA practice

For simplicity, the results of the review are mostly displayed and discussed in an aggregated form, meaning that generally no study is singled out unless it is relevant for illustrating the presented arguments and recommendations. Likewise, the aggregation of the studies over time ignores the development in LCA competence over the last decades. LCA practice in the mid-90s thus presents several flaws when evaluated with the current methodological state of LCA. Although these old flawed studies should not be considered as 'poor' as recent studies with the same flaws, such time-related biases were disregarded in the present review, i.e. all studies were treated equally in the review process. However, it is worth noting that more than 50% of the studies in the review have been published since 2009 (Laurent et al., 2013). Therefore, the existence of time-related biases is believed to be of minor influence in the interpretation of the past LCA practice.

The past LCA practice was compared against widely-accepted reference works, namely the LCA standards ISO 14044 (ISO, 2006), the ILCD Handbook guidelines (EC, 2010a) and their related technical guide applied to the waste sector (EC, 2011). The ILCD Handbook guidelines, which heavily rely on the ISO standards 14040 and 14044, are the result of the analysis of a "large number of LCA manuals of business associations, national LCA projects, consultants and research groups as well as scientific LCA publications" and have been subject to extensive consultations from experts and stakeholders in industries, authorities and academia (EC, 2010a). Therefore, these scientifically-based guidelines can be considered sufficiently representative to be taken as a reference work in the analysis of the past LCA practice. The 222 studies were reviewed for each major methodological step within the goal definition, scope definition, life cycle inventory, life cycle impact assessment and result interpretation of the LCA (complete list of reviewed elements in Appendix A of Laurent et al., 2013). These major steps are illustrated in Fig. 1, which also serves as a rough outline that may ease the reading of Sections 3–7; readers are thus encouraged to refer to it, whenever needed. Not all aspects of Fig. 1 are addressed in this paper (omitted aspects are in italic types). For keeping the structure of the paper as simple as possible, some cross-cutting aspects appearing in several steps, e.g. multi-functional processes handled in both scope definition and life cycle inventory (LCI) analysis, are addressed in one single section of the paper; their relevance in other sections is however indicated.

2.3. Presentation of the review results

The analysis of the results is presented in Sections 3–8 following the LCA methodological steps described in Fig. 1. Out of the 222 reviewed studies, only a few were ISO-compliant studies having undergone an external peer-review (e.g. Clauzade, 2010; Grant et al., 2001, 2003; Jenseit et al., 2003; Kreißig et al., 2003; Shonfield, 2008; Fisher et al., 2006). Apart from Clauzade et al. (2010), all of these are public reports. Approximately 20% of the 222 studies stated that they followed the ISO14040/44 standards in their assessments. The reference or not to the ISO standards cannot be considered as an indicator of the level of knowledge of the LCA methodology. However, it is worth noting that several studies claiming to follow the ISO standards were found to include critical flaws in their application of the methodology (data not shown). In contrast, with a very few exceptions (e.g. Blengini et al., 2012), nearly none of the studies quoted the ILCD Handbook guidelines (EC, 2010a, 2011), which seem too recent to have been applied in published case studies yet.

3. Goal definition

3.1. Overall consideration

The ISO standards outline the goal definition as the part framing the intended uses and users of the LCA case study based on its overall context description (ISO, 2006). The ILCD Handbook guidelines divide the goal definition in 6 aspects, viz. the intended applications, the limitations to usability of results, the drivers and

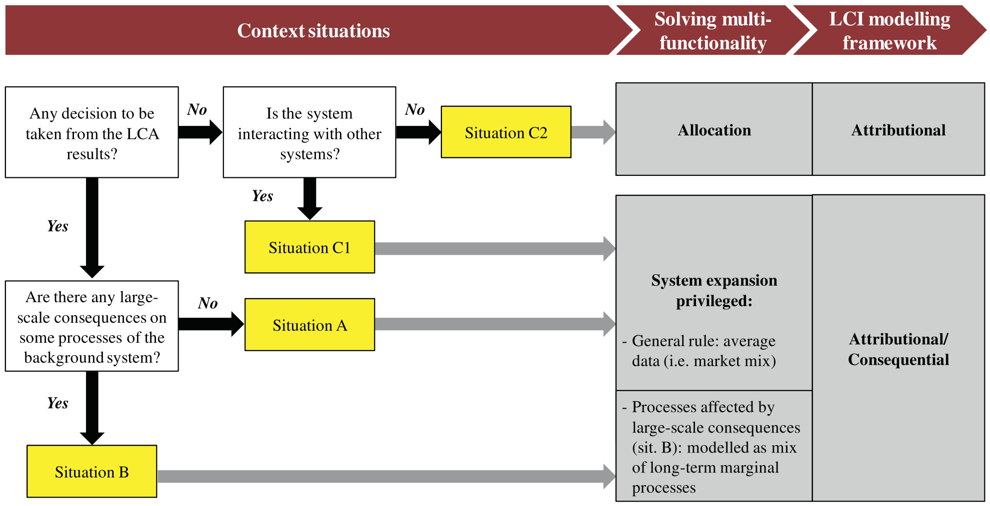

Fig. 2. Identification of context situations and LCI modelling framework (derived from EC, 2010a).

motives, the target audience, the potential disclosure to the public, and the commissioner of the study (EC, 2010a). However, several of the reviewed studies are LCAs with no other purpose than supporting methodological developments or specific discussions. As a result, an adequate definition of the goals of the study, as in, e.g., Arena et al. (2003), Jenseit et al. (2003) and Blengini et al. (2012), is missing in most reviewed studies. The authors of studies either skip this step or limit their definition of goals to briefly describing the actual work performed, e.g. 'comparing different integrated systems of mixed waste management with regard to their environmental performance' or 'evaluation of environmental burdens associated with the municipal waste management in a municipality'. No information about the context and the possible uses of the LCA study are thus indicated.

The goal definition has strong implications for the further conduct of the LCA study, in particular the scope definition. It is a determining factor in the selection of the LCI modelling framework (see Section 3.2). It also frames the interpretation of the results, i.e. what can be concluded from the results and, most importantly, what cannot. Scientific articles are often the result of research projects or initiatives, which typically are not commissioned by any entity. It may justify the observed weak identification of the intended uses and users in most studies. On the other hand, such practices may lead the readers to misinterpret or misunderstand the reach and limitations of the conclusions and recommendations. Examples include LCA studies assessing a limited number of relevant impact categories and claiming the environmental superiority of one waste management alternative over another without nuances (e.g. Gunamantha and Sarto, 2012), or LCA studies generalising comparative results, which are strongly dependent on the local conditions (see Laurent et al., 2013). Therefore, although the goal definition of an LCA can be adapted depending on its background, e.g. commissioned ISO-compliant LCA study versus research-support LCA case study, the authors are recommended to provide sufficient information on the context of the study (see also Section 3.2) and the usability of the results, including the limitations of the LCA to prevent misinterpretation of its results.

3.2. Context situations

According to the ILCD Handbook guidelines, a system under assessment can be split into two main components, namely the foreground system, which includes processes specific to the analysed system, e.g. own operations, and the background system, which includes processes that are not specific to the analysed system, e.g. generation of electricity which is purchased via an assumed homogeneous market (EC, 2010a). The ILCD Handbook defines four major types of context situations, namely the situation A (micro-level decision support), the situation B (meso/macro-level decision support), and the situations C1 and C2 (accounting with no decision support) – see also Fig. 2. They are dependent on the intended decision implications of the study as well as on either the existence of large-scale consequences on some processes in the background system and in other systems (differentiation of situations A and B), or the existence/consideration of interactions of the system with other systems (differentiation of situations C1 and C2, the latter being very rare) – see decision tree in Fig. 2.

A proper identification of the context situation is important because it determines the type of LCI modelling framework to apply (see Fig. 2 and Section 4.2 for specification of LCI modelling framework), which has a considerable influence on the results and their interpretation. For example, the use of either allocation or system expansion in an attributional modelling can lead to opposite results (e.g. wood waste treatment in Werner et al., 2007).

However, none of the reviewed studies has undertaken the identification of context situations as recommended in the ILCD Handbook guidelines. Based on a context analysis of the 222 reviewed studies, all context situations, including the accounting/monitoring-type context situations C (e.g. Waeger et al., 2011; Niskanen et al., 2009), were estimated to be represented although a majority of the studies could potentially lead to decisions (e.g. assumed the case when authors provide recommendations on which systems or routes are environmentally preferable). Several studies seem to be micro-level assessments (situation A), in which specific solid waste treatment routes are investigated in a given municipality (e.g. Ortiz et al. (2010) for construction waste; Chen et al. (2011a,b) for plastic waste; Hong et al. (2006) for municipal waste; Aye and Widjaya (2006) for organic waste; Birgisdottir et al. (2007) for bottom ashes; Tonini and Astrup (2012) on residual waste; Johansson et al. (2008) for sewage sludge, etc.) or in a given facility or company (e.g. Munoz and Navia, 2011; Rivela et al., 2006; Navia et al., 2006; Damgaard et al., 2011; Manfredi et al., 2009; Scipioni et al., 2009). However, without knowledge of the context and whether or not some processes have large-scale consequences on installed capacity of the background system (e.g. energy market at national scale), the differentiation between context situations A and B is difficult to make.

To identify context situations in future studies, the practitioners are thus recommended to follow the decision tree from Fig. 2, which comes down to answering two questions. In the event of an identified context situation B, the practitioner is additionally advised to identify all the processes affected by large scale consequences because they are modelled using mixes of long-term marginal processes (see Section 4.2). Finally, in open-loop recycling systems, in which the secondary good replaces a different kind of material, energy or parts (e.g. incinerated plastics leading to energy recovery, ultimately resulting in electricity used in other applications), and for which situation A has been identified, the practitioners are strongly recommended to evaluate the existence of a sufficient capacity for the alternative processes or systems delivering functionally equivalent energy or materials (EC, 2010a). If this is not the case, i.e. the market cannot absorb the changes, the study is a situation B and these processes should be modelled as mixes of long-term marginal processes (EC, 2010a).

4. Scope definition

4.1. Functional unit and reference flows

The review of the studies led to identifying four major classes of functional unit (f.u.), namely (i) unitary f.u., defined by a unitary measure, e.g. management of 1 tonne of waste, (ii) generation-based f.u., defined by the waste generation in a delimited region for a specified period of time, (iii) input-based f.u., defined by the waste amounts entering a given facility, and (iv) output-based f.u., defined by the waste by-products, e.g. amounts of recovered energy or recycled material. The distribution of these f.u. types, illustrated in Fig. 3, shows that by far most LCA studies of SWMS use a unitary functional unit, e.g. management or treatment of 1 tonne of waste. This indicates that practitioners often define the functional unit with the intent to give a round reference flow or that they simply mistake the meaning of the functional unit with that of the reference flow. The functional unit gives a quantitative description of the primary function fulfilled by the systems under study and is a guarantee of their comparability whereas the reference flow expresses the physical flows required by the systems or scenarios under study to fulfil the functional unit, i.e. to provide the same service. Therefore, a functional unit reduced to a unitary measure, e.g. "1 tonne of municipal waste", is in fact a reference flow that does not contain sufficient information as to the function of the system since it does not specify the type of waste considered. For example, 1 tonne of municipal waste in Aarhus is not comparable to 1 tonne of waste in Beijing. When assessing a municipal solid waste management system, the waste composition can thus be expected to be specified. As another example, if the assessment focuses on alternatives for managing plastic waste, the types of plastics considered would be required in the event that one of the compared systems is not able to handle a specific type. Therefore, in addition to explicitly reporting the functional unit (note the 17% of studies, which did not specify any functional unit; see Fig. 3), the practitioners are encouraged to include these local specifications as well as any relevant aspects to guarantee the comparability of the systems.

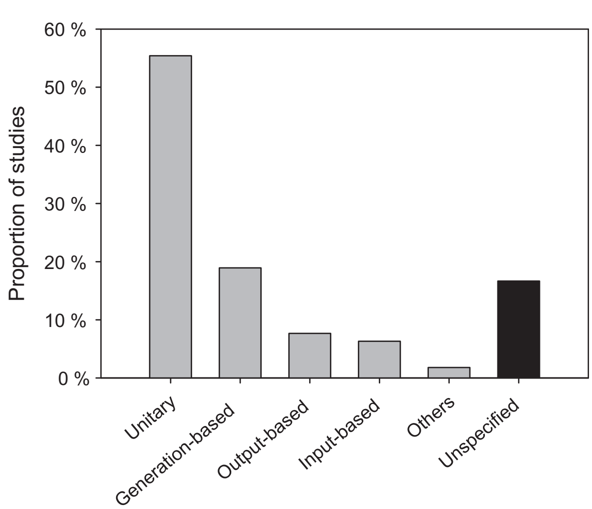

Fig. 3. Proportions of studies for each class of functional unit (total of 222 studies). A number of studies are classified into more than one category, including studies, for which the functional unit was not explicitly reported ("unspecified") but could be retrieved from the displayed results, e.g. graph captions.

4.2. Applied LCI modelling framework and issues of multi-functional processes

The LCI modelling framework describes the modelling approach that has been adopted for solving the multi-functionality of some processes, i.e. the point that some processes provide more than one function. The incineration of mixed waste with energy recovery or the recycling processes are such examples (both providing the service of treating the waste and delivering a commodity as output). Multi-functional processes can be addressed by two fundamentally different approaches – by attributional or consequential modelling (EC, 2010a). The choice of the LCI modelling framework needs to be justified in the scope definition of an LCA study, but the details pertaining to the identification of the multi-functional processes and the methods to perform system expansion and allocation should be documented in the LCI analysis phase. For simplification purposes, all discussions on those aspects are centralised hereafter.

As introduced in Section 3.2, the choice is strongly dependent on the contextual situation of the LCA study although other aspects may be influential, e.g. practical feasibility, reproducibility or stakeholder acceptance (EC, 2010a). In Fig. 2, practitioners can find a basis to identify their LCI modelling framework from the contextual situation of their study. As a general guidance, the ILCD Handbook recommends the use of system expansion with average data for situations C1, A and B although processes of the background system and/or other systems that are specifically affected by large-scale consequences should be modelled using a mix of long-term marginal processes (see Fig. 2).

Because of the evident confusion of concepts and terminology surrounding the types of LCI modelling framework, all imaginable types of modelling could be found in the past practice – see Fig. 4. These range from the incomplete recognition of multi-functional processes, through the mixing of approaches, e.g. with the use of system expansion for energy recovery together with the allocation for material recovery, to a consistent selection of either system expansion or allocation. In the application of system expansion, discrepancies also arise with respect to the choice of the marginal or average data, which often appears to be arbitrary.

To address multi-functional processes, system expansion has mostly been applied (ca. 75%) while allocation has been exclusively used in ca. 4% of the reviewed LCA studies (Fig. 4). Among the latter group of studies, different allocation keys were used, such as mass (e.g. Lundie and Peters, 2005), heat value (e.g. Jenseit et al., 2003; Chen et al., 2011a,b), waste volume (e.g. Grant et al., 2001), exergy (e.g. Cherubini et al., 2008) and economic value (e.g. Chen et al., 2011a,b; Shen et al., 2010); some studies also applied the cut-off approach defined by Ekvall and Tillman (1997) (e.g. Shen et al., 2010; Martinez-Blanco et al., 2010). The conduct of system expansion or allocation is the source of controversy in the LCA community (Finnveden et al., 2009). The different allocation procedures,

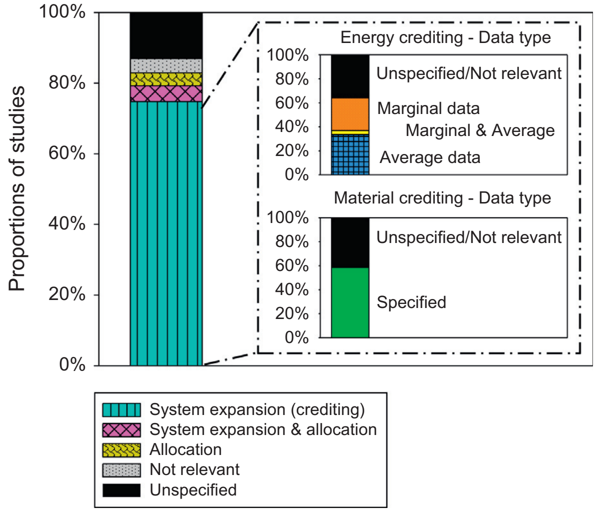

Fig. 4. Handling of multi-functional processes (total of 222 studies). Energy includes both heat and electricity. The differentiation between marginal and average data for material crediting is largely omitted in the studies, and is hence not reported in the figure.

as exemplified above, can lead to different results, and no international consensus as to which one(s) to apply in practice has been reached yet (e.g. Finnveden, 1999). On the other hand, system expansion has been criticised for being largely uncertain due to the number of assumptions to estimate what is avoided (e.g. Heijungs and Guinee, 2007). However, most of the reviewed studies having performed allocation are not compliant with the "ISO hierarchy" that recommends prioritising first the subdivision of processes and then the system expansion over the use of allocation, which should be used as a last resort (ISO, 2006). This prioritisation is also supported by the ILCD Handbook guidelines, which additionally links it to the goals of the study, i.e. via the identified context situations (EC, 2010a).

To perform system expansion for crediting materials recovery, virgin production processes are often considered the most appropriate choice (see, e.g., Frees, 2008). However, none of the studies, albeit marked as "Specified" in Fig. 4, addresses the selection between marginal or average data. This lack of transparency is believed to arise from the difficulties in identifying the appropriate approach and arguing for the type of material sources to be replaced (e.g. Finnveden, 1999). LCA practitioners are strongly advised to abridge this gap in their LCA studies. As indicated in Section 3.2, the selection of the type of data is dependent on the context situations specific to the study. In some situations, e.g. open-loop recycling, an analysis of the market mechanisms may be necessary to ensure a proper reflection of the consequences of the analysed system. A number of guidelines have been provided in the literature for consequential modelling, e.g. Ekvall (2000) or Ekvall and Weidema (2004). Regardless of this issue, most reviewed studies assume a substitution ratio set to 1:1 and/or a quality similar to the substituted product (e.g. Shen et al., 2010; Björklund et al., 1999; Briffaerts et al., 2009; Le Borgne and Feillard, 2001; Marinkovic et al., 2010; Rigamonti et al., 2009a) although a few studies account for a quality decrease in the materials recovered using different assumptions (e.g. Bernstad et al., 2011; Rigamonti et al., 2009b), sometimes including a differentiation of the approaches per type of materials recovered (e.g. Munoz et al., 2004). An overestimated substitution ratio or grade of the recovered materials can significantly impact the benefits gained from recycling and ultimately play an important role in the final LCA results. Therefore, in relevant cases, where the substitution ratio is not accurately known, the practitioners are recommended to consider the necessity of including this as part of a sensitivity analysis.

With regard to energy crediting, Fig. 4 shows that a relatively equal amount of studies used either national grid mix or marginal energy supply for crediting the supply of electricity and heat out of the system (e.g. from incineration). However, like for the crediting of materials recovery, there is an important problem of transparency in the studies. While approximately 25% of the studies (data not shown) do not specify the type of data they used for crediting energy recovery, most of the others do not provide justifications for selecting either average or marginal data. Exceptions that could serve as good examples to practitioners in their future studies include Lundin et al. (2004), Bergsdal et al. (2005), Bernstad and la Cour Jansen (2012), Boughton and Horvath (2006), Hospido et al. (2005) and Merrild et al. (2012) (non-exhaustive list of examples).

In addition to the studies not documenting their selection of data for crediting materials and energy recovery, it is alarming that ca. 13% of the studies did not report the identification and handling of multi-functional processes (see black bars in Fig. 4). Provided the large influence of these aspects on the outcome of the LCA (e.g. Shonfield, 2008; Werner et al., 2007), practitioners are thus strongly recommended to transparently document (i) the arguments pertaining to their selection of LCI modelling framework (see also Section 3.2), and (ii) the details associated with the data and assumptions used for handling of multifunctional processes.

4.3. System boundaries

The system boundaries define which processes in the life cycle are included or excluded from the assessed system (ISO, 2006; EC, 2010a). The system delimitation needs to ensure that all relevant processes, and hence their potential environmental impacts, are included in the assessment. Ultimately, an accurate definition of the system boundaries contributes to reducing the risk of burden-shifting from one part of the life cycle to another. As indicated in the ILCD Handbook (EC, 2010a, 2011), the selected type of LCI framework modelling has a significant influence on the definition. Whereas attributional modelling describes the system as following a process-chain logic, consequential modelling includes all consequences that decisions may have on the background system and/or other systems, hence expanding the extent of processes to be included.

In the following, based on the reviewed past practice, three specific aspects in the setting of system boundaries are separately analysed: (1) capital goods, (2) collection and transportation, and (3) transport and treatment of secondary products and final residuals. A concluding section additionally addresses the reporting of the system boundaries. Fig. 5 illustrates the past practice with respect to these three aspects.

4.3.1. Capital goods

Capital goods used in the treatment chain, e.g. construction and decommissioning of facility and machineries, were included in 12% of the studies (see Fig. 5). In 26% of the studies, authors have excluded capital goods justifying this choice – typically without references – as being of little importance to the overall result. Even more concerning are the 62% of the studies, which omit to address the exclusion or inclusion of capital goods. The definition of capital goods is also unclear across studies. For instance, the construction of landfills is regarded as such in Bientinesi and Petarca (2009) and Menard et al. (2004) whereas it is not in Boughton and Horvath (2006).

A recent series of studies investigating the importance of capital goods, i.e. buildings and machineries, demonstrated the relative importance of including capital goods in waste management systems. Although the disposal stage of the capital goods was not in-

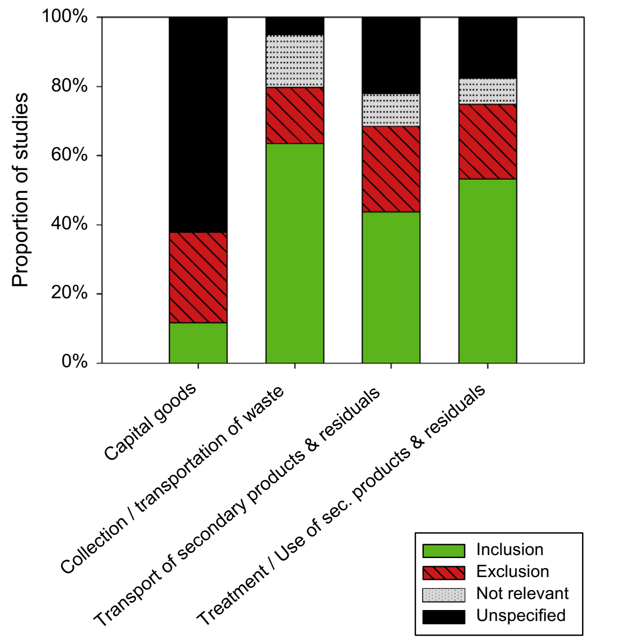

Fig. 5. Inclusion and exclusion of some aspects in/from system boundaries (total of 222 studies).

cluded, contributions of up to 85% to several impact categories were found for collection and transport systems (Brogaard and Christensen, 2012); lower but non-negligible contributions to climate change for incineration and landfilling (estimated 2–3% and 1–8%, respectively) and to abiotic resource depletion for landfilling systems were also reported (Brogaard et al., 2013a,b). Considering an entire management system of municipal waste, the contribution of capital goods to certain impact categories may thus not be negligible. These results are also consistent with the more generic findings reported in Frischknecht et al. (2007), who showed that capital goods had significant contributions to abiotic resource depletion, climate change and ecosystem damages in many waste management system (LCI processes from Ecoinvent). Based on the detailed knowledge of the different processes, technologies and waste management routes included in their analysed system, practitioners are thus recommended to examine with care the potential contribution of capital goods before deciding to exclude them.

4.3.2. Collection and transportation

Processes of collection and transportation of waste were included in ca. 63% of the studies (Fig. 5). In 16% of the studies, authors had chosen to exclude such processes. Common justifications for exclusion are the assumed irrelevance of these processes based on the outcome of previous studies and their identical use across the compared scenarios (e.g. Hong et al., 2010; Lee et al., 2007; Zhao et al., 2009a). Such arguments are in agreement with the ISO standards (ISO, 2006), although great caution should be exercised. In several studies, the environmental relevance of waste collection and transportation in the assessed SWMS was demonstrated to be insignificant, e.g. the increase of transport/collection in recycling systems not offsetting the benefits of recycling over that of incineration with recovery (e.g. Merrild et al., 2012; Salhofer et al., 2007). Yet, these conclusions cannot be generalised because most of these studies are only site-specific and because other studies, albeit in minority, have brought nuances in their conclusions due to the possible influence of transport distances, collection schemes and the emission standards met by the vehicles (e.g. Beigl and Salhofer, 2004; Larsen et al., 2009). Therefore, instead of commonly referring to earlier studies for disregarding collection and transportation processes, a case-specific evaluation is recommended to practitioners.

4.3.3. Secondary products and final residuals

Secondary products, defined as valuable outputs (e.g. digestate or compost), and waste treatment residuals (e.g. air pollution control (APC) residues or residues from material recycling and wastewater treatments) are here considered together because the boundary between the two can be very context-dependent. For example, bottom ashes can be landfilled or re-used in road construction depending on the common practice in a considered region (e.g. Birgisdottir et al., 2007). Analysing the past practice (see Fig. 5), processes related to the transport and treatment or utilisation of both secondary products and treatment residuals are only included in 44% and 53% of the studies, respectively. In ca. 22–25% of the studies, these aspects were not addressed (e.g. Boldrin et al. (2011) and Cadena et al. (2009) in relation to wastewater treatment; Merrild et al. (2012), Boldrin et al. (2011), Dahlbo et al. (2007), Dodbiba et al. (2007), Larsen et al. (2010), Manfredi et al. (2011), Nishijima et al. (2012) and Riber et al. (2008) in relation to incineration ashes; Martinez-Blanco et al. (2010) in relation to impurities from industrial composting, etc.). Some exclusions could however be justified in the studies, e.g. Merrild et al. (2012) arguing the negligible impacts from treatment of incineration ashes over a 100-year time horizon shown in Birgisdottir et al. (2007) and Hyks et al. (2009).

Therefore, the practitioners are recommended to adopt a case-specific approach and evaluate the potential environmental relevance of including such processes in relation to the context of the analysed SWMS. Likewise, although they have been disregarded in several studies (e.g. Cabaraban et al., 2008; Kaplan et al., 2009; Tabata et al., 2011), the valorisation of secondary products, such as compost or digestate used as soil amendment, can alter the LCA results by their additional environmental benefits. In addition, the generation of these outputs could in many cases be interpreted as secondary functions, which, to be compliant with the ISO standards (ISO, 2006) and ILCD Handbook guidelines (EC, 2010a), should be addressed following the requirements of the selected LCI framework modelling (see Section 4.2).

4.3.4. Reporting of system boundaries

In addition to the recommendations specific to each above aspect of the system boundaries (Sections 4.3.1–4.3.3), the practitioners are strongly advocated to document inclusions and exclusions of processes in a transparent manner. This is typically lacking in studies although it is important for the readers to be aware of what the LCA results actually do account for and what they do not. The ILCD Handbook recommends the use of a semi-schematic diagram explicitly representing the parts of the life cycle that are considered or not (EC, 2010a). Although most studies comply with this requirement, it is not a systematic practice. To report the different assumptions pertaining to the inclusion or exclusion of processes in a comprehensive manner, the use of Supplementary Information or Appendices in scientific articles should also be considered by practitioners (for good examples, see, e.g., Blengini et al., 2012; Cook et al., 2012; Scharnhorst et al., 2006).

4.4. Impact coverage

The ISO standards and ILCD Handbook recommend to include all relevant impact categories and to report and justify any limitations with regard to the impact coverage of the assessment in the scope definition (ISO, 2006; EC, 2010a). Fig. 6 illustrates the past practice with respect to inclusion of impact categories. Less than 50% of all studies performed complete assessments in either of

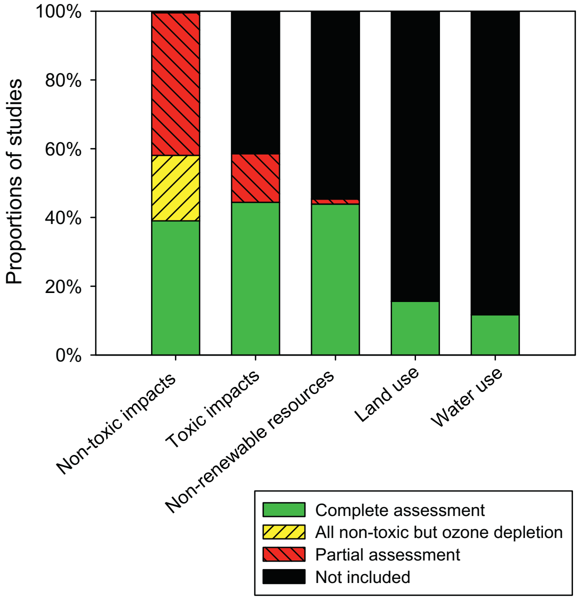

Fig. 6. Proportions of impacts covered in the assessments (total of 222 studies). Non-toxic impacts include climate change, stratospheric ozone depletion, acidification, photochemical ozone formation, eutrophication. Toxic impacts include aquatic and terrestrial ecotoxicity, human toxicity and impacts from particulate matters.

the 5 defined groups of impact categories. Considering the coverage of the entire environmental burden, only 29% of the 222 studies assessed both non-toxic and toxic impact categories in a relatively comprehensive manner. Although they are marked as "complete assessment" under the category "toxic impacts" in Fig. 6, several studies often rule out one aspect of ecotoxicity impacts and few studies thus encompass both terrestrial and aquatic ecotoxicity-related impacts. In the studies conducting partial assessment of toxicity, 93% have given priority to impacts on human health, thus disregarding impacts on ecosystems.

Authors arguing for such exclusion or truncated inclusion of toxic impacts often raise the uncertainties associated with the methods for assessing (eco)toxicity impacts. These arguments may have been acceptable in the early 2000s. However, with the recent developments in characterisation modelling of toxicity-related impacts, i.e. the consensus model USEtox (Hauschild et al., 2008a; Rosenbaum et al., 2008, 2011; Henderson et al., 2011), and the on-going research for quantifying waste-specific toxicity-related impacts, these arguments are no longer valid. Numerous studies have demonstrated the significant impacts on human health and ecosystems caused by releases of chemicals, particles or pathogens from waste management systems (e.g. Vergara and Tchobanoglous, 2012; Giusti, 2009; Hauschild et al., 2008b). These evidences should thus compel practitioners to systematically include the assessment of toxic impacts to prevent potential burden-shifting (Laurent et al., 2012).

As indicated in Fig. 6, the assessment of resources, which embraces the assessment of land use, freshwater use and/or depletion of non-renewable resources like metals and fossils, was also dismissed in several studies. However, their assessment may not have been relevant for all of the studies. Non-renewable resource depletion was addressed in about half of the studies although its relevance might be high in many studies, e.g. in situations involving recycling of materials or energy recovery. There is however a current absence of consensus in its impact assessment as reflected by the lack of agreed definition of the area of protection to be represented and the consequent large variety of existing indicators, e.g. exergy-based indicators versus scarcity-based indicators (EC, 2010b; Hauschild et al., 2013). Having been less assessed (each accounting for less than 15% of studies; Fig. 6), land use and freshwater use also suffer from a lack of operational and comprehensive impact assessment methods. However, progress toward consensus has recently been made for freshwater use (Kounina et al., 2013; Tendall et al., 2013) and new methods for assessing impacts of land use have recently emerged (e.g. impacts on biodiversity in de Baan et al. (2012); impacts on biotic production in Brandao and Mila i Canals (2012)). Land use may not be relevant in all assessments of SWMS, but its relevance in systems heavily dealing with biomass and sewage sludge has been raised (e.g. Dahlbo et al., 2007; Poulsen and Hansen, 2003). For example, systems including recycling of wood products may be expected to avoid deforestation and hence a loss of biodiversity; the application of sewage sludge on land is also expected to change soil quality, and hence facilitate plant growth. The relevance of including water use assessment may also be dependent on specific situations, e.g. where contaminants from landfill reach groundwater used for human consumption or irrigation. Such cause-effect pathways are still not covered consistently in life cycle impact assessment (LCIA) although attempts have been made to use indicators quantifying "spoiled groundwater resources" (e.g. Birgisdottir et al., 2007; Manfredi and Christensen, 2009; Manfredi et al., 2010). Because the inclusion of non-renewable resource depletion, land use and freshwater use assessments can be strongly dependent on the context and characteristics of the SWMS under study, the practitioners are therefore advised to analyse their relevance carefully before ruling them out.

5. Life cycle inventory analysis

The life cycle inventory analysis is the phase, which builds on the requirements defined in the goal and scope phases to conduct the collection of data on flows to and from the processes of the waste management system, the further data handling to reach a comprehensive emission and resource consumption inventory, and the modelling of the analysed system (EC, 2010a). In typical LCA studies, this phase is the most time- and resource-demanding for LCA practitioners.

To ease its reading, Section 5 is divided into four parts. It first gives an introduction to the major challenges associated with the building of inventories for SWMS that also account for their local specificities. A second sub-section characterises the past practice with respect to the data collection, addressing the data sources, the achieved data quality and the data representativeness. The use of software to facilitate the modelling of the system is then discussed in a third part, prior to a concluding sub-section on the reporting quality of the studies with respect to transparency and reproducibility of the inventory.

5.1. Modelling context-specific inventories

Compared to traditional product LCAs, the application of LCA to SWMS brings challenges because the results are often dominated by a few waste treatment processes for which the local conditions strongly affect their impacts and hence need to be accurately modelled. Four major aspects were identified as being of high relevance and these are separately analysed in the following in relation to the approaches undertaken by practitioners:

- The heterogeneity of the waste composition, which is strongly dependent on the local conditions.
- The different qualities of the material flows, which depend on the source separation and the applied waste collection schemes.

- The need for tracking trace pollutants, which is dependent on the final destination of the processed waste.
- The inventory of long-term emissions, which stem from specific uses or disposals of processed waste, e.g. landfilling.

5.1.1. Describing waste compositions

In general, waste compositions were found to be poorly described in the reviewed studies. In several cases only few details are presented, only at the level of products, without detailed composition of the waste fractions. The level of details required should be adapted to the scope of the study and enable reproducibility of the results. For example, in the assessment of biological treatment alternatives, the contents of C, N and P are important to specify, and in some cases, so is the potential for methane formation as these aspects will have a strong impact on the possibilities for nutrient and energy recovery (e.g. Hospido et al., 2005; Bernstad and la Cour Jansen, 2011). When assessing incineration processes, the specification of the lower heating value of the waste is similarly important as it determines the energy output that could be credited (e.g. Chen et al., 2011a,b). Finally, a last example could be the importance of specifying the content of heavy metals in assessments that cover toxicity-related impact categories (e.g. Fruergaard et al., 2010).

In addition to the general lack of refined waste compositions, the data sources used for those that are reported are usually not sufficiently detailed. For example, in some studies, the waste compositions originate from a specific sampling campaign (e.g. Gunamantha and Sarto, 2012; Bernstad et al., 2011), while in others, they are derived from earlier studies, reports or databases (e.g. Koroneos and Nanaki, 2012). Only few studies described the uncertainty and variability of the waste composition, while several studies have shown that these uncertainties lead to important uncertainties in the LCA result (e.g. Slagstad and Brattebø, 2013). Waste compositions are specific to each case study and are dependent on geographical differences and the type of SWMS considered (e.g. see Koneczny and Pennington, 2007; Turconi et al., 2011).

Therefore, LCA practitioners are recommended to (i) refine their waste compositions not only at a product level but also at a content level, and (ii) report the data used in a transparent and comprehensive manner. The use of Supporting Materials or Appendices can serve this purpose; this can be correlated with the planning of a sensitivity analysis testing different parameters of the waste composition (e.g. see Andersen et al., 2012; Chen and Christensen, 2010; Hanandeh and El-Zein, 2010; Tonini and Astrup, 2012).

5.1.2. Modelling waste collection

Waste collection, including the source separation, can generally be considered a key element in the outcome of most LCA studies because it is located upstream of any further treatment, utilisation and disposal of the waste (Larsen, 2009). Only a proper description of the underlying waste composition, collection schemes and sorting efficiencies can thus guarantee a proper modelling of the waste flows in these downstream processes (Larsen, 2009).

In that setting, the consumers' behaviour can have a large influence on the composition of the separately collected fractions and on that of the residual waste. For example, the occurrence of missorting, which is typically not modelled in the studies, can have a strong impact on the quality of processed waste in subsequent treatments (e.g. Bernstad et al., 2011). With regard to residual waste, Rigamonti et al. (2009a) considered changes in its composition through three different scenarios that included separate collection of materials and food waste with collection rates of 35%, 50% and 60%. The authors showed that the potential for energy recovery and the associated emissions from the residual waste incineration could change significantly over these scenarios (Rigamonti et al., 2009a). In addition to the consumers' behaviour, the collection schemes have an important influence on the LCA results. Examples can be a decreased moisture content in incinerated waste following the introduction of a separate collection and treatment of organic waste; an increased moisture content resulting in a decreased heating value of the incinerated waste following an increased material recycling (particularly paper and plastics); changes in heating values and methane production when more or new fractions of recyclables are recycled.

As a consequence, LCA practitioners are strongly encouraged to secure a deep knowledge of the collection system, including the source separation. This knowledge is a pre-requisite to accurately map the mass flows, assess their potential variability across the different analysed alternatives, and ensure a consistent inventory modelling. To bring more robustness, a complementary analysis of the influence of source-separation and missorting can also be performed through a sensitivity analysis – see Section 7.2. Examples of studies having undertaken such step are Aye and Widjaya (2006), Rigamonti et al. (2009a), Bernstad et al. (2011) and Boldrin et al. (2011).

5.1.3. Tracking trace pollutants

Trace pollutants, e.g. small quantities of heavy metals, need to be consistently tracked through the SWMS because they may result in important damages, e.g. via thermal treatment (e.g. Fruergaard et al., 2010). One approach can be the use of transfer coefficients, as applied for example by Hellweg et al. (2001) and Fruergaard et al. (2010) on different compounds in incinerated waste to assess the composition of the different incineration residues (e.g. flue gas). Applications to other contexts than incineration are also relevant, e.g. accumulation of hazardous compounds in recycled materials (i.e. risk-cycling). To help track the different substances, the practitioners should consider the use of material and substance flow analysis (MFA/SFA) – see e.g. Björklund and Bjuggren (1998) or Lederer and Rechberger (2010).

5.1.4. Inventorying long-term emissions

Long-term emissions are slow emissions occurring over centuries or millennia, which taken as a whole may constitute a substantial environmental burden (Hauschild et al., 2008b; Doka and Hischier, 2005). Their modelling is a particular challenge since no measurements exist and only poorly validated, predictive models can thus be used. Three types of situations are addressed in the following, i.e. (1) the degradation of organic matter in landfills, (2) the leaching of pollutants from landfills or use-on-land (UOL) and (3) the nutrient substitution when applying processed organic waste on land. The modelling of these processes is generally not documented in the reviewed studies although a few examples could be retrieved.

The degradation of organic matter in landfills can be modelled at different levels of parameterisation. Some inventories of emissions are based on emission rates per tonne of waste (e.g. Munoz and Navia, 2011), while others rely on more refined properties like the differentiation between carbon structures (cellulose, lignin etc.) as performed in Koroneos and Nanaki (2012). Emission rates can be based on methane potentials of the different fractions (e.g. Manfredi and Christensen, 2009), or on a first-order decay model (e.g. Kaplan et al., 2009). Moreover, the possible collection of the produced landfill gas needs to be modelled, as well as its possible utilisation or oxidation in the top cover. The modelling of those processes can be particularly relevant for assessing climate change impacts of the SWMS (Manfredi and Christensen, 2009).

When waste is deposited on land or in a landfill, the leaching of pollutants obeys different geochemical laws. The leachability of the different compounds in the waste should be modelled to quantify

the amounts of pollutants escaping the landfill in a given time horizon. Most studies disregard this process, as it is deemed too uncertain. However, it means that the landfills are effectively modelled as free sinks for pollution, thus ignoring the potentially large impacts from toxic compounds (e.g. Doka and Hischier, 2005). Different modelling approaches can be adopted to estimate the leaching. For example, Koroneos and Nanaki (2012) employed a first order differential rate equation, while Manfredi and Christensen (2009) used compositions defined at different time periods. Two approaches currently exist with regard to modelling the time horizon for long-term emissions – see Section 6.2.

Finally, the modelling of the application of processed organic waste on agricultural land is as complex a task as the two previous situations. While UOL generates emissions of carbon and nitrogen compounds, it also avoids other emissions which would have occurred with the use of mineral fertilizers. These processes depend on many regional parameters such as the type of soil, the climate, the legislation, the behaviour of farmers and the type of crop cultivated. For example, Sonesson et al. (2000) – and generally all other studies using the ORWARE model – use a model for nitrogen turnover in soil, which calculates the increased emissions of nitrate and ammonia caused by using organic instead of mineral fertiliser. The loss of nitrogen to water and air is then calculated as a function of the amount of organic nitrogen, ammonium and nitrate spread combined with information on spreading techniques, soil type and yearly rainfall.

From the three above examples of long-term emission situations, it can be observed that despite their high relevance, efforts are still required to reach consistent and comprehensive modelling approaches (see also Section 6.2). For those 3 aspects, practitioners are thus required to be transparent with respect to their modelling assumptions and their potential influence on the final results.

5.2. Data collection and representativeness

5.2.1. Data sources and quality

Fig. 7a, which provides an overview of the data sources used, shows that about 70% of the studies use site-specific data although many of these studies refer to already published works containing these data. This use of site-specific data typically goes together with the use of literature sources (ca. 77%) and/or databases (ca. 66%). In many cases, the sourcing of the original data set, i.e. the actual time represented by the data and not the year of their publication, is not documented by the studies. Such lack of transparency is very common with the handling of primary data, where many papers (re)use data presented in other LCA studies without further explanations than just citing the study. It also applies to the handling of secondary data for which it is very difficult to know the sources and characteristics of used data, e.g. when authors only refer to databases without specifying the year of data collection. Tracking the original data sources of the studies could help provide an overview of the data representativeness in the LCA studies (see Section 5.2.2), but this was not possible within the current study. Instead the cited literature of each study was roughly evaluated based on the publication years and considered regions, thus reflecting a "best-scenario" assessment (see details in Appendix A of Laurent et al., 2013). Approximately 29% of the studies were found to use inadequate literature data, either out of date or inconsistent with the region under study (Fig. 7a). This supports the general assessment of data quality which is shown in Fig. 7b, in which about one third of the studies was found to be of good data quality while ca. 22% were found of poor quality, e.g. due to unique and improper use of literature for the study (see background framework, incl. criteria, for the data quality assessment in Appendix A of Laurent et al., 2013).

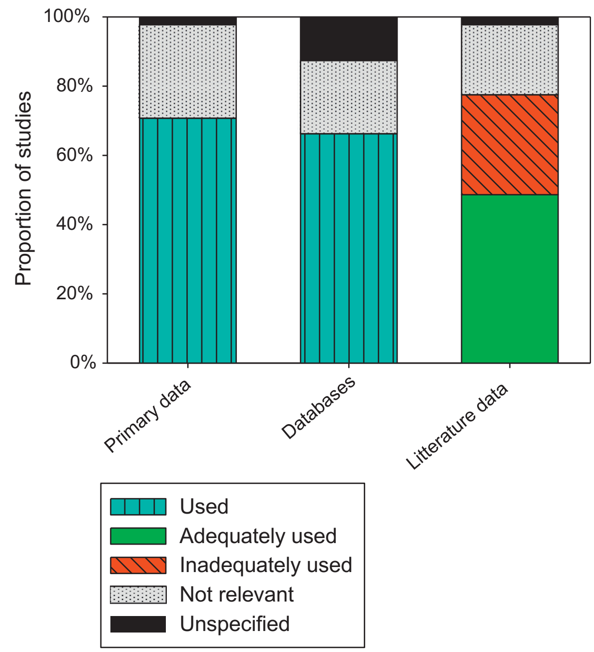

Fig. 7a. Types of data sources used in studies (total of 222 studies). Primary data refers to site-specific data from e.g. field investigations. Databases mainly refer to LCA software-embedded databases. Adequacy of literature data was based on a rough consistency check with the temporal and geographical scopes of the study (see details in Appendix A of Laurent et al., 2013).

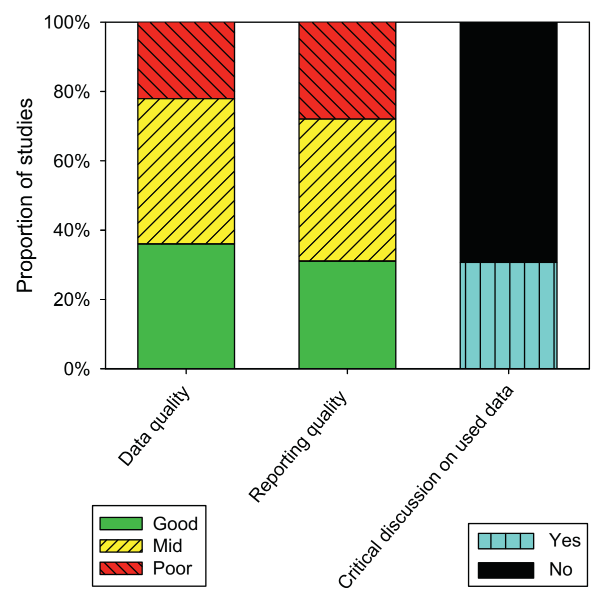

Fig. 7b. Evaluation of 3 major criteria in the building of the inventory (total of 222 studies). Data quality was evaluated based on used data sources and types; the reporting quality was evaluated based on transparency and reproducibility (e.g. of assumptions) in both data collection and handling; the existence of a critical discussion of the used data (data appropriateness) considered the existence of either qualitative or quantitative discussion (see details in Appendix A of Laurent et al., 2013).

These findings demonstrate one of the major hurdles in the application of LCA to SWMS, i.e. the access to up-to-date, site-specific data to build an inventory that matches the scope of the analysed system. As indicated in Section 5.1, the local specificities of a SWMS, e.g. waste compositions, are important to capture in the

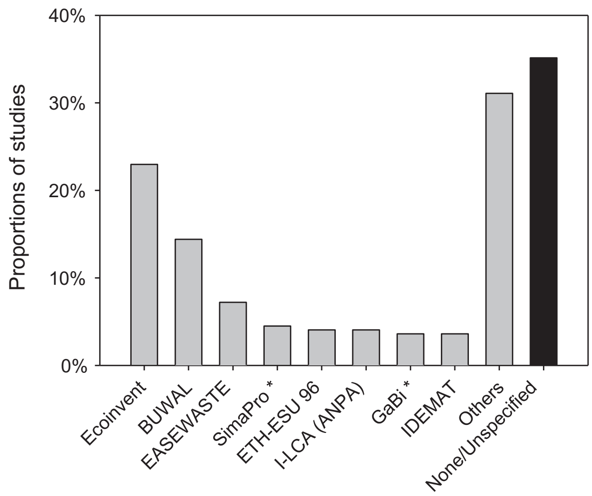

Fig. 7c. Databases used in studies (total of 222 studies). Others include International Data Bank Boustead Ltd., Australian LCA db, Franklin USA 1998 and, in general, all LCI databases embedded in waste-specific LCA software (see Fig. 8). Some authors simply referred to the software containing the databases (marked with "*") whereas they are not database providers (PE actually is, but authors referred to databases embedded in GaBi software without further specifications).

inventory modelling because they often determine the outcome of the LCA study. However, they require data, which are not always available, sometimes even for key processes in the analysed system. Practitioners thus need to find the most appropriate 'data proxies'. The use of inventory models (see examples in Section 5.1) and waste-specific LCA software (see Section 5.3) could be helpful in that context. Although primarily used for modelling the background system, several databases, e.g. ecoinvent and BUWAL, are also used to complete the inventory (see Fig. 7c).

5.2.2. Data representativeness

Addressing the geographical, time and technological representativeness of the inventory consists in characterising how well the modelled inventory captures the processes, which relate to the actual life cycle of the SWMS. This step is important to consider because the local specificities of the analysed SWMS can have a large influence on the LCA results (see Laurent et al., 2013), and thus should be well captured.

In 31% of the studies, the practitioners included a critical discussion of the data sources and subsequent quality of the data while constructing their inventory (see Fig. 7b). Although bending the ISO requirements and ILCD Handbook guidelines, which advocate the matter to be addressed as early as in the scope definition (ISO, 2006; EC, 2010a), this could easily be defended from a practical point of view because the paucity of data typically compels practitioners to adjust their scope from the availability of data – instead of the recommended opposite. In these 31% of studies, several authors thus acknowledge the lack of representativeness but still justify their use of such data by the absence of alternative data sources (e.g. Dahlbo et al., 2007; Coelho and de Brito, 2012; Nakatani et al., 2010; van Haaren et al., 2010). Most of the LCA studies included in this review were performed in the course of projects, in which financial support and hence time availability may have been limited, thus preventing a thorough collection of data. However, because of the potentially large implications on the LCA outcome, practitioners are encouraged to devote efforts to (i) iteratively evaluate and refine the representativeness of their inventory in relation to the defined scope of the study, and (ii) quantify its influence on the LCA results through the systematic use of uncertainty/sensitivity analyses on identified key parameters – see Section 7.2.

Pedigree matrices (Weidema and Wesnaes, 1996) could thus be used to identify critical aspects with regard to data sources and handling (e.g. Scipioni et al., 2009; Boughton and Horvath, 2006). With regard to the geographical appropriateness, the most concerning issues arise in regions, where LCA has not been much applied (see Laurent et al., 2013), thus compelling practitioners to use waste data and LCI processes from other regions, e.g. Ecoinvent LCI processes typically matching Swiss situations. This might introduce large biases in the results, e.g. in relation to landfill or biological treatment, even though some studies attempted to adapt such imported data or processes to the local conditions (e.g. Cook et al., 2012; Zhao et al., 2009b). With respect to time appropriateness, because the original data sets are not always tracked to the sources (see Section 5.2.1), practitioners tend to overlook the actual age of the data used, and hence rarely address this aspect. As an example to reduce the lack of time representativeness, the inventory could include forecasting or backcasting projections of key parameters, e.g. waste generation (Bergsdal et al., 2005; Assamoi and Lawryshyn, 2012; Miliute and Kazimieras Staniskis, 2010). As of technological appropriateness, which strongly relates to the geographical and time dimensions, it may become highly relevant when emerging technologies are compared with commercially available technologies. Data on developing technologies are commonly gathered from pilot plants or even laboratory works, which are often associated with much higher environmental burden than commercial scale operation. Up-scaling data into full-scale commercial conditions is an alternative although it may be difficult and uncertain (e.g. see Jenseit et al., 2003; Assefa et al., 2005; Chevalier et al., 2003).

5.3. Software used for modelling

To model the analysed SWMS, different software tools were used by practitioners. Not all studies are transparent in that respect (see black bar in Fig. 8). Analysing the distribution of the software used in the reviewed studies, it is observed that several studies used generic LCA software like SimaPro and GaBi, while a similar share used dedicated waste system modelling tools such as EASEWASTE, ORWARE, WRATE or WAMPS. Over the years, a large variety of dedicated waste-LCA software has been developed, typically designed to be applied in the countries where they were

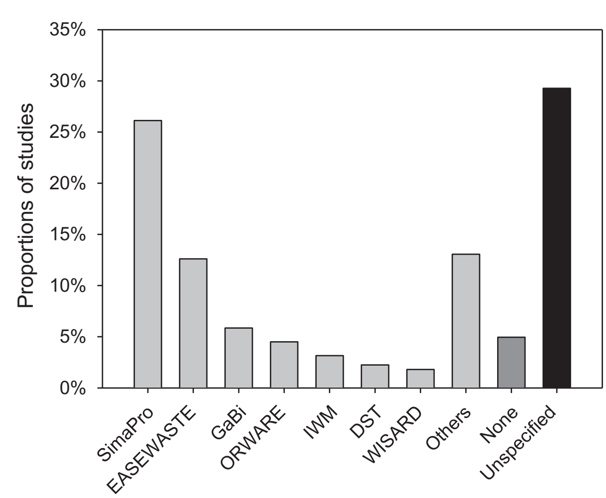

Fig. 8. LCA software used in studies (total of 222 studies). The category "Others" includes TEAM, TRACI, UMBERTO, GEMIS, WRATE, LCAiT, JEMAI-LCA, EIME, WAMPS software.

developed. A review study of the different assumptions embedded in these software has demonstrated that LCA results are independent from the choice of software as long as the models are based on the same assumptions while it has also highlighted the need for harmonisation of some of the technical modelling aspects, e.g. non-geographic assumptions like time horizons for landfill emissions (Gentil et al., 2010).

In the modelling of SWMS, we recommend the use of dedicated waste-LCA software over generic ones because they can bring several advantages by providing adapted frameworks to model the systems and by tackling the different issues introduced in Section 5.1. Some software can thus allow a consistent tracking of the materials and their associated properties for the different waste flows, include the modelling of waste-specific processes, e.g. the embedment of an organic matter degradation as a first order decay model (e.g. in the ISWM-DST model; Kaplan et al., 2009), or support mass balance checks for individual waste components. In addition, several of these software embed waste-specific databases, which may be representative enough for the analysed SWMS with regard to e.g. waste composition and LCI processes. However, although waste-specific LCA software are overall demonstrated to be consistent between each other, they typically have different levels of refinements. Therefore, practitioners should be careful to select one that can capture with sufficient accuracy the complexity of the analysed SWMS in relation to its defined scope.

5.4. Reporting of LCI analysis

The transparency and reproducibility of an LCA study are critical aspects that need to be respected when performing an LCA (EC, 2010a). However, as indicated in Fig. 7b, the reporting quality of the reviewed studies with respect to the LCI analysis phase was found to be acceptable in only ca. 31% of them while nearly as many were deemed of poor quality. Particularly in scientific articles, the documentation of data sources, the critical assessment of data appropriateness and/or the details on the assumptions and modelling performed to construct the inventory are often incomplete. Particular care should thus be exercised in future works. Blengini et al. (2012), Waeger et al. (2011), Rivela et al. (2006) and Cook et al. (2012) are examples of good reporting practice for practitioners (non-exhaustive list). Because of limited space in scientific articles, the use of Supporting Information, as done in most of these examples, is an efficient means to ensure a good reproducibility and transparency in the LCA study.

6. Impact assessment

6.1. Past practice

CML (31%), EDIP (21%) and Ecoindicator 95 or 99 (EI95/99; 14%) are the most widely used LCIA methodologies to assess SWMS (see Fig. 9). As indicated in Section 4.4, only 29% of the reviewed studies have included both non-toxic and toxic impact categories in their assessment – most of them being in fact studies using single-score endpoint indicators such as EI95/99. Most studies stopped at the characterisation step while 46% of the studies performed normalisation and 26% performed weighting (a majority due to the embedded use in EI95/99). It is also worth noting that ca. 20% of the studies have omitted to report which LCIA methods or characterisation models were used in their assessments. In some of these cases, authors simply omit to describe which LCIA method or characterisation model they used. In others, they only cite the LCA software which they used, presupposing that the reader knows which LCIA method or

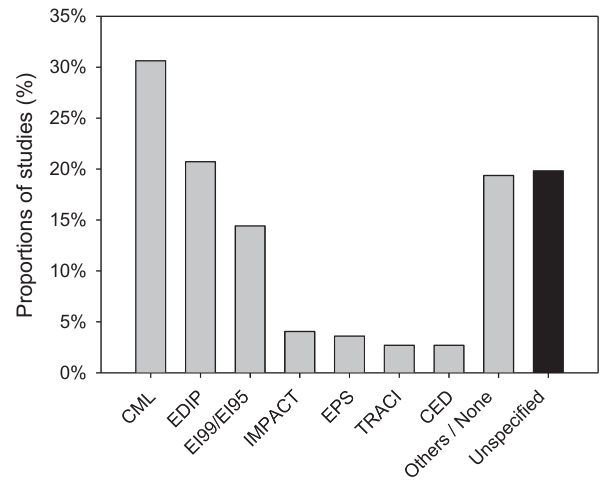

Fig. 9. LCIA methods used (total of 222 studies). Some studies used more than one LCIA method; all have been included here. Category "Others" includes the use of specific models, which are not considered as whole LCIA methods, e.g. IPCC (2007).

characterisation model is embedded in it. However, most of the widely used LCA software from Fig. 8 include several operational LCIA methods, hence preventing the identification of the one applied in the study. Considering the potentially large influence that the selection of the LCIA methods may have on the results (e.g. Dreyer et al., 2003; Pizzol et al., 2011a, 2011b), such lack of transparency and reproducibility should be excluded from common practice.

Although it is an integral part of the interpretation step, through the completeness, consistency and sensitivity checks (EC, 2010a), a number of studies also addressed uncertainties related to the impact assessment phase. Most of the raised aspects are addressed qualitatively and large variations in the extent of the discussion exist between the studies (see also Section 7.2). Most frequently raised LCIA flaws relate to the incomplete impact coverage regardless of the inclusion or not of the toxic impacts, the general assessment of toxic impacts, the land use assessment, the biogenic carbon or C sequestration accounting approach, the assessment of long-term emissions, the regionalisation of impact assessment to adapt local conditions, the weighting approach and the need for including other indicators than environmental impact indicators (data not shown). The first three aspects are covered in Section 4.4 in relation to the selected impact coverage, which should be specified in the scope of the study. A number of the remaining aspects, which still constitute methodological challenges, are developed in Section 6.2.

6.2. Methodological challenges

Several methodological challenges are general LCIA issues with high relevance in the application to SWMS, e.g. occupational health impact characterisation, while others are unique to the assessment of SWMS, e.g. characterisation of impacts from long-term emissions. A number of these impediments, which may alter the results of an LCA study, are described hereafter. The list is not intended to be exhaustive, but simply aims at providing an eye-opener to LCA practitioners, who should be aware of some of the important flaws in current impact assessment methods and integrate them, when interpreting their results, in their decision-support to stakeholders.

The biogenic carbon accounting is an aspect often considered a source of uncertainties (see Section 7.2). However, a careful modelling should prevent the occurrence of mistakes. As concluded by Christensen et al. (2009), biogenic CO_2 emissions can be seen as neu-

tral or contributing to climate change, as long as there is consistency both throughout the specific waste treatment system and between the compared systems. For example, if a 100-year timeframe is used in the estimation of landfill gas generation, the biogenic carbon remaining in landfills after the end of this period should be accounted for to maintain the mass balance (Moberg et al., 2005). In addition, the allocation of none or negative global warming potential to the remaining biogenic carbon should be the same as in other waste treatment alternatives, e.g. carbon remaining in biological fertilisers and ashes/biochars from thermal treatments after 100 years.

As indicated in Section 5.1, long-term emissions of heavy metals from landfills are particularly relevant in LCA applied on solid waste management systems. Although a consistent and parsimonious methodology to account for their impacts is currently lacking, a number of approaches have been developed (Doka, 2009; Finnveden et al., 1995; Hauschild et al., 2008b; Hellweg et al., 2003; Obersteiner et al., 2007; Pettersen and Hertwich, 2008). In common LCA practice, 2 major groups of approaches can be identified (Doka, 2009): (1) the discard of all emissions occurring after a limited time period (typically 100 years), and (2) the accounting of all emissions considering an infinite time horizon. However, both lead to results that are hard to accept for many decision makers. In the former, most emissions are simply cut off (since ca. 99% of heavy metals remain in the landfill after 100 years), whereas, in the latter, the resulting toxic impacts tend to completely dominate the entire environmental profile (Hauschild et al., 2008b). The lack of consensus led the studies addressing this issue (ca. 5%) to either include all emissions over a long time period (approach 2; e.g. Scharnhorst et al., 2006), use a combination of both (e.g. Zhao et al., 2009b), or, as often the case, disregard the toxic impacts from heavy metal emissions (e.g. Mendes et al., 2004; Del Borghi et al., 2007; Nakakubo et al., 2012). Such disarray shows the need to develop an adequate method to assess toxic impacts from long-term emissions of heavy metals. Until a consistent, operational method becomes available, the LCA practitioners should be aware of the potential biases that may exist in their results, and integrate this risk in their interpretation.

Although not raised in the reviewed studies, other more generic methodological issues are worth mentioning here because of their relevance to LCA of SWMS. Due to the large variety of situations, which could lead to exposure of workers to toxic agents, the characterisation of occupational health impacts is one of them. For example, waste collectors have been demonstrated to be affected by exposure to dust, pathogens and chemicals in both developed and developing countries (e.g. Giusti, 2009; Breum et al., 1997; Poulsen et al., 1995; Nielsen et al., 1997; Chi et al., 2011). To date, no applicable occupational exposure model exists to account for those agents, in particular dust and pathogens. The assessment of the loss of resources may also include potentially relevant uncertainties in situations, where these impacts play a major role, e.g. biowaste treatment or recycling systems. As indicated in Section 4.4, it currently suffers from a lack of methodological consensus. Finally, the regionalisation of the impact assessment methods can become an important contribution to strengthen the LCA studies applied to SWMS in capturing local conditions and providing a context-specific support to decision-makers. Promising outcomes in that direction are currently being released from various projects, e.g. LC Impact (http://www.lc-impact.eu/) or IMPACT World+ (www.impactworldplus.org/).

7. Interpretation

7.1. Results interpretation

In solid waste management, the strength of LCA stems from its ability to quantify context-specific environmental impacts and improvement potentials taking into account local specificities in the modelling of the waste treatment systems. Further discussion on the main findings and their interpretation in a selection of reviewed studies can be found in Laurent et al. (2013). According to the ILCD Handbook guidelines (EC, 2010a), the LCA results should be interpreted in relation to the goals and scopes of the study, which, for most reviewed studies, are not always clearly defined (e.g. see Section 3.1). Therefore, a large variety of interpretations can be found, ranging from the mere report of the impact scores obtained, through the ranking of the different alternatives, to the thorough analysis of the results in their context. In addition to interpreting the results in relation to the context of the LCA study, the practitioners are also encouraged to deepen their interpretations of the results, e.g. identification of most contributing substances and processes to each impact, systematic consideration of the uncertainties in a quantified form (see Section 7.2), and identification of improvement potentials, on which decision-makers should focus to maximise the environmental benefits. The interpretations should also account for the completeness, consistency and sensitivity checks (EC, 2010a). In particular, the sensitivity check can be helpful in identifying the most influential parameters of the SWMS – see Section 7.2. Examples of good practice that can serve as inspiration to thoroughly interpreting the results are Cook et al. (2012) and Fisher et al. (2006).

7.2. Uncertainty and sensitivity analyses

Referred to as “sensitivity check” in the ILCD Handbook guidelines (in addition to the completeness and consistency checks; EC, 2010a), these analyses are diverse but can be presented in 2 groups: sensitivity analysis and uncertainty propagation. Sensitivity analysis investigates how results are influenced by the model inputs while uncertainty propagation aims at quantifying the overall uncertainty of the results. Both are separately presented in the following. Table 1 presents an overview of the types of analyses performed in the 222 studies.

Sensitivity analysis methods are used to assess how results are sensitive to variations in input data and to modelling choices. Scenario analyses are one-factor-at-a-time (OFAT) methods serving

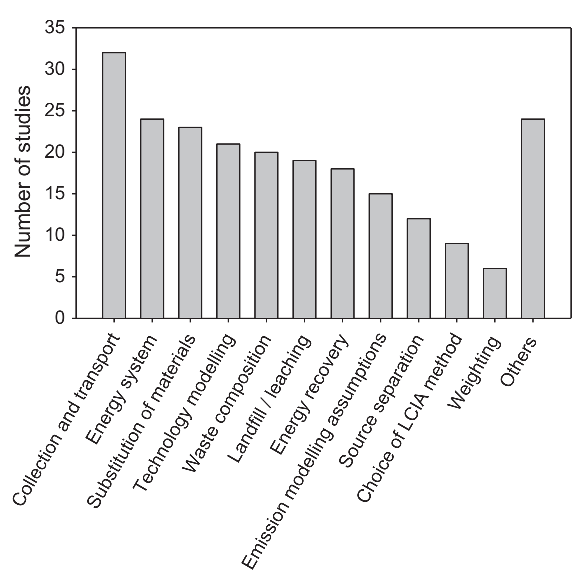

Fig. 10. Issues covered in sensitivity analysis (total of 101 studies). The category “Others” (25 studies) includes carbon accounting methods (6), inclusion of secondary materials (4), allocation rules (4), accounting of waste containers (3), time horizon in impact assessment (3), testing of other treatments (2), choice of databases (2) and normalisation (1).

Table 1
Overview of past practices for uncertainty and sensitivity analyses in LCA studies of solid waste management systems (total of 222 studies).

Practices[a]		Number of studies
Sensitivity analysis method	Uncertainty propagation method	
Scenario analysis	–	96
Scenario analysis + perturbation analysis	–	1
–	Monte Carlo analysis	4
Scenario analysis	Monte Carlo analysis	2
Uncertainty contribution analysis	Monte Carlo analysis	3
Scenario analysis	Taylor series	2
–	Rough calculation of uncertainty	1
–	Fuzzy sets	1
Special case: reporting of ranges of results		1
Not addressed or only qualitative discussion included		111

[a] Some studies used either sensitivity analysis methods or uncertainty propagation methods alone (rows marked with a "–"). Others used a combination of both.

the main purposes of investigating the robustness of the results and identifying the most sensitive parameters that could influence the results of the LCA and hence ultimately affect the recommendations provided to decision-makers. A number of 101 studies performed a scenario analysis, comparing original results with results obtained when using different data and/or assumptions. The number of tested assumptions varied a lot across the studies; Fig. 10 shows the most tested key parameters. The parameters for collection and transportation were the most commonly tested although studies typically concluded that their influence on the final results was limited. Due to the uncertainty and controversy surrounding the types of energy and materials that are actually substituted in system expansion (see Section 4.2), several studies have made these parameters vary under different assumptions. This practice can still be recommended because markets for recycling and energy systems are likely to quickly evolve when considering time horizons of 10–20 years that are relevant to most LCA of SWMS. Waste composition was the fifth most tested parameter. Since the heterogeneity of the treated materials often appears to be a significant source of uncertainty, the practitioners are strongly recommended to include waste composition in sensitivity analyses (see also Section 5.1). Finally, it can be worth combining a sensitivity analysis with a qualitative assessment of the data uncertainties (e.g. Boldrin et al., 2010) as this enables to assess the criticality of a parameter based on the combined analysis of its sensitivity and its uncertainty.

Less popular than sensitivity analysis, uncertainty propagation was only addressed in 13 out of the 222 reviewed studies. Twelve of them propagated probability distributions by analytical or stochastic methods while one study used fuzzy sets. These practices are in accordance with the observations of Lloyd and Ries (2007), who reviewed the application of uncertainty analyses in LCA. The results of the uncertainty propagation were used to (i) test the robustness and reliability of results (Cook et al., 2012; Hanandeh and El-Zein, 2010; Ciacci et al., 2010; Hong and Li, 2012; Pires et al., 2011); (ii) support the recommendations provided to decision-makers, i.e. demonstrating the absence of benefit in the studied improvement scenario (Hong and Li, 2011), identifying the improvement potentials in the parts of the modelling (Scipioni et al., 2009) and providing ranges of results (Grant and James, 2005); and (iii) contribute to methodological developments, i.e. introducing fuzzy normalising and weighting factors (Guereca et al., 2007). The level of details and quality of the uncertainty analyses vary considerably among the studies. However, some trends could be identified. It is a common difficulty to represent the uncertainties of the input data which will be propagated in the model. To select the probability distributions, different methods were used: expert judgment and literature (Cook et al., 2012; Hanandeh and El-Zein, 2010; Pires et al., 2011; Suh and Rousseaux, 2002), pedigree matrix as introduced by Frischknecht et al. (2005) (Scipioni et al., 2009; Hong and Li, 2011), a combination of the two (Grant and James, 2005) or a simplified approach based on Weidema and Wesnaes (1996) (Ciacci et al., 2010). One study did not provide documentation on their selections (Hong and Li, 2011). It was observed that the studies estimating the uncertainty distributions of their data reported their assumptions better than the studies adopting the pedigree matrix approach, where the assumed uncertainty distributions were not presented. Additionally, authors estimating their data uncertainties typically adopted uniform distributions, sometimes justified by the fact that no information about the uncertainty distribution shape was available, while the pedigree matrix approach imposes the choice of lognormal distributions. Cook et al. (2012) is a good example of how transparent the process of uncertainty propagation should be. The choice of the distribution shape and values is explained for each of the 62 parameters included in the uncertainty propagation. In addition, following the uncertainty propagation, the authors performed a sensitivity analysis to quantify the contribution of each parameter to the overall uncertainty. Finally, they developed a digested, easy-to-read support to communicate the results of the uncertainty propagation, i.e. by showing the ranges of values between 25th and 75th percentiles, and those of the sensitivity analysis, i.e. by summing up the most influent parameters for all impact categories and their 3 scenarios (Cook et al., 2012).

As recommendations for performing sensitivity and uncertainty analyses, the practitioners are referred to the guidance provided in Clavreul et al. (2012), who developed a four-step tiered approach adaptable to different needs and resources. As a preliminary step the users are advised to screen all results using contribution analyses so that the less significant impact categories and processes can be disregarded in the rest of the analysis. The first and mandatory step consists in assessing the sensitivities of the LCA results to all main assumptions by scenario analysis. A list of the uncertainties typically encountered in waste-LCAs is provided in Clavreul et al. (2012). With regard to parameter values, it is advised to run a perturbation analysis. This method, described by Heijungs and Kleijn (2001), extends the use of the scenario analysis by ranking all parameters according to their influence on the results. Secondly, if resources allow, uncertainty propagation should be performed, while keeping a focus on providing clear and justified assumptions, so that the results of the uncertainty propagation can be meaningful. Thirdly, a sensitivity analysis can be performed on the results of the uncertainty propagation. Clavreul et al. (2012) provide a simplified method that allows to estimate the contribution of each parameter by using results of the perturbation analysis. As a fourth step, it is suggested to provide a visualisation of the shift in the ranking of different options due to variations of two most critical key parameters.

Table 2
Overarching recommendations for a better LCA practice in LCA studies of solid waste management systems.

Goal definition
• Consider it in all studies, including scientific articles
• Specify context, intended use and limitations of the usability
• Identify context situation according to ILCD Handbook guidelines –use Fig. 2
Scope definition
• Functional unit: Specify all elements that characterise the function of the system and its specificities, e.g. waste compositions for municipal solid waste
• LCI framework analysis: (i) Select the framework in accordance with the goals of the study and the ILCD Handbook guidelines (use Fig. 2); (ii) Transparently document assumptions and type of data choices, e.g. selection of marginal/average data for the crediting of energy and materials
• System boundaries: (i) Evaluate carefully the relevance of including capital goods, waste collection/transportation processes and transport/treatment of secondary products and waste treatment residuals; (ii) Transparently delimit and document all inclusions and exclusions with adequate justifications
• Impact coverage: Include all relevant impact categories, including toxic impacts and potentially non-renewable resource depletion, land use and freshwater use depending on context
Life cycle inventory analysis
• Collect representative data and build an inventory that captures all relevant local specificities of the analysed solid waste management systems. A specific focus should be laid on (i) the modelling of waste flows and compositions at the beginning of the analysed system, e.g. source separation and waste collection schemes, because they govern the flows in the downstream processes; (ii) the modelling of processes at the end of the system, e.g. landfilling or energy recovery processes, because potential benefits of the waste management systems typically originate from these processes; and (iii) the modelling of diffuse emissions, e.g. long-term emissions from landfilling, because they can be important contributors to impact potentials
• Document the data collection process as well as the inventory building assumptions and modelling in a transparent and reproducible manner
• Consider the use of dedicated waste-specific LCA software as they enable tracking of the different waste flows and their properties, checking of mass balances for individual waste components, and modelling of waste-specific environmental mechanisms. Their built-in LCI processes may be sufficiently representative for parts of the analysed system to avoid collection of specific data
Life cycle impact assessment
• Document the LCIA methods used and be aware of the underlying flaws and uncertainties in impact assessment methods, e.g. impacts from long-term emissions
Life cycle interpretation
• Make a thorough analysis of the impact potentials, e.g. using contribution analyses at substance and process level, and identify improvement potentials in relation to goals and scope of the study (context, usability limitations)
• Systematically couple the analysis of the results with sensitivity checks, combining sensitivity analysis and uncertainty propagation, to identify the most critical key parameters of the analysed system, and check mass balances and flows for the most important flows

8. Recommendations

Through the review of the 222 LCA studies of SWMS, a number of methodological points were identified as problematic in the current practice because of their lack of compliance with the ISO standards and the ILCD Handbook guidelines. In most cases, this lack of consistency may have significant influence on the results of the study and, ultimately, alter the recommendations provided to the different stakeholders. Therefore, it is important that LCA practitioners ensure that their application of the LCA methodology, also accounting for the challenges specific to the application of LCA to SWMS, does not jeopardize the validity of this support. In parallel, it is the role of the peer-reviewers (e.g. in scientific journals) to carefully check that such practices are effectively implemented in case studies. To assist both practitioners and reviewers in their respective 'duties', recommendations were provided within each step of the LCA methodology phases, i.e. goal definition, scope definition, life cycle inventory analysis, life cycle impact assessment, and life cycle interpretation. LCA practitioners are referred to each of the corresponding sections in the present paper to find the detailed recommendations. As a non-exhaustive summary, Table 2 highlights the main aspects of these recommendations.

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